Life Histories, Salinity Zones, and Sublethal Contributions of Contaminants to Pelagic Fish Declines Illustrated with a Case Study of San Francisco Estuary, California, USA

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Abstract Human effects on estuaries are often associated with major decreases in abundance in aquatic species. However, remediation priorities are difficult to identify when declines result from multiple stressors with interacting sublethal effects. The San Francisco Estuary offers a useful case study of the potential role of contaminants in declines of organisms because the waters of its delta chronically violate legal water quality standards; however, direct effects of contaminants on fish species are rarely observed. Lack of direct lethality in the field has prevented consensus that contaminants may be one of the major drivers of coincident but unexplained declines of fishes with differing life histories and habitats (anadromous, brackish, and freshwater). Our review of available evidence indicates that examining the effects of contaminants and other stressors on specific life stages in different seasons and salinity zones of the estuary is critical to identifying how several interacting stressors could contribute to a general syndrome of declines. Moreover, warming water temperatures of the magnitude projected by climate models, increase metabolic rates of ectotherms and can hasten elimination of some contaminants. However for other pollutants, concurrent increases in respiratory rate or food intake result in higher doses per unit time without changes in the contaminant concentrations in the water. Food limitation and energetic costs of osmoregulating under altered salinities further limit the amount of energy available to fish; this energy must be redirected from growth and reproduction toward pollutant avoidance, enzymatic detoxification, or elimination. Because all of these processes require energy, bioenergetics methods are promising for evaluating effects of sublethal contaminants in the presence of other stressors, and for informing remediation. Predictive models that evaluate the direct and indirect effects of contaminants will be possible when data become available on energetic costs of exposure to contaminants given simultaneous exposure to non-contaminant stressors.

Keywords Susceptibility to toxins, Bioenergetic costs, Impaired waterways, Multiple stressors, Pelagic organism decline, Climate change

Introduction

Urbanization and agricultural development around estuaries are often associated with major changes in the abundance and composition of aquatic species (Nichols et al. 1986; Chaudry and Zwolsman 2008). Such shifts in estuarine communities coincide with altered hydrology, which affects salinity and turbidity, as well as inputs of municipal waste water, fertilizers, and pesticides. A common challenge to prioritizing mitigation is that biota are responding to an array of stressors that each have sublethal effects. Contaminants may be important contributors to death by a thousand cuts, but the difficulty of quantifying their independent effects makes the often high economic cost of regulating individual stressors such as pesticides seem excessive and unfair when other factors such as waste water output from municipalities also contribute. The resulting societal demand for unequivocal proof of appreciable mortality hinders progress toward remediation.

Even when waters are legally impaired according to a regulatory standard, it is often hard to attribute declines of a variety of species to contaminants on the basis of standard toxicological tests that use mortality of a few well-studied species as the main criterion of effect. The desire to generate legal criteria that address contaminants in isolation from confounding factors has led to reliance on standardized laboratory tests with limited environmental reality (Heugens et al. 2001). However, ecological effects of single or multiple contaminants, like the influences of weather, disease, food shortage, or behavioral disturbance, are more commonly sublethal contributors to a suite of adverse ecological conditions (Porter et al. 1984). Moreover, exposure to contaminants along with other stressors is often highly episodic, as after major storms or

during blooms of toxic cyanobacteria. Such periodic events can greatly affect populations, but can go undetected by rigidly scheduled environmental monitoring of weekly or monthly habitat quality.

In fish, sublethal effects can occur as subtle or cryptic impairments such as altered behavior or suppressed immunity, which are difficult to measure even under controlled laboratory conditions. In rivers and streams or tidal systems, detecting such effects requires direct observation of fish impairment and immediate collection of biological and water samples before the toxicant moves away from the sampling location. A solution to the difficulty of measurement may lie in recognizing that energy is redirected from growth and reproduction toward avoiding, detoxifying, or eliminating pollutants, and that energy demands increase when contaminant exposure is coupled with other stressors such as limited food or altered salinity (Beyers et al. 1999a; Beyers et al. 1999b). Thus, measuring energetic costs by methods such as comparing growth rates or reproductive output to that of control organisms captures the impact of single and compounding stressors, and logistically, is more feasible than measuring cryptic effects of contaminants.

In estuaries, salinity and turbidity are key habitat parameters, and use of specific habitat zones varies among species and life stages depending on their osmoregulatory capability (Feyrer et al. 2010). In this paper, we present a conceptual model of the susceptibility to contaminants of four fish species with different habitat-use patterns (anadromous, brackish, and freshwater) in the San Francisco Estuary. In this scheme, we consider season, location, and mode of action of contaminants for each life stage. We then review published evidence of sublethal but statistically significant effects of contaminants (including inorganic nitrogen) on fish or their food in the San Francisco Estuary and elsewhere. We consider interactions among contaminants as well as with

125 environmental factors (e.g., salinity, turbidity). We also explore how warming water
126 temperatures associated with projected climate change may increase or decrease toxicity of
127 contaminants depending on their mode of action. Finally, we highlight emerging research aimed
128 at evaluating how sublethal stressors affect energy costs rather than mortality per se, and thereby
129 address what may be the most practical measure of contaminant effects at the community level.
130 Quantifying energy costs at the level of individuals may reveal energy deficits that lead to
131 population declines; however, data collection and monitoring are generally not geared toward
132 energy costs. Our goal is to recast thinking about contaminant effects and criteria for identifying
133 them, and to encourage development of suitable methods of assessment.
134
135 The case study of San Francisco Estuary
136
137 Around the year 2002 in the San Francisco Estuary (California, U.S.A.), long-term declines in
138 the abundance of multiple fish species accelerated abruptly, even though surrounding land use
139 was stable and several years of above-average precipitation were expected to improve the
140 reproductive success of many of the fishes (Sommer et al. 2007; Thomson et al. 2010). The San
141 Francisco Estuary (henceforth "the Estuary") consists of San Francisco Bay (most seaward
142 region, including San Pablo Bay), Suisun Bay (intermediate, brackish waters), and the freshwater
143 Sacramento-San Joaquin Delta (henceforth "the Delta") at the confluence of the Sacramento and
144 San Joaquin rivers (Fig. 1). Declining species include native delta smelt (*Hypomesus*
145 *transpacificus*), which is listed as endangered under the U.S. and California endangered species
146 acts, and longfin smelt (*Spirinchus thaleichthys*), which is listed as threatened under the

147 California Endangered Species Act. Introduced threadfin shad (*Dorosoma petenense*) and young
148 of the year (age-0) striped bass (*Morone saxatilis*) have also declined (Fig. 2) .
149 The Delta is designated as an impaired waterway under Section 303(d) of the U.S. Clean
150 Water Act, meaning that the allowable total maxima of daily loads of some contaminants are
151 chronically or repeatedly exceeded. To evaluate biological effects of water quality, researchers
152 have collected waters from various locations across the entire Delta. After exposing organisms to
153 those water samples under controlled laboratory conditions, researchers documented widespread
154 toxicity to invertebrates, planktonic algae and larval fish during times when the four fish species
155 of interest are in the Delta (Fig. 1). The percentages of toxic samples range from 0-15%
156 depending on test species, sampling frequency, and location, which is highly heterogeneous
157 among projects (Werner et al. 2000; Johnson et al. 2010; Werner et al. 2010). Despite such
158 evidence of the presence of toxic chemicals in Delta waters, fish kills clearly associated with
159 contaminants have not been documented since the 1980s. The criterion of overt mortality among
160 wild populations as evidence of effect may overlook the contribution of chronic contaminant
161 exposure to declines caused by sublethal impairment and the cumulative effects of multiple
162 stressors.
163
164 Life histories of declining fishes
165
166 Delta smelt are endemic to the brackish and fresh waters of the Delta (Moyle 2002; Bennett
167 2005) (Fig. 3). The species has a one-year life cycle. Spawning occurs in the spring in the
168 freshwater portions of the Delta, although the exact characteristics of spawning habitat have not
169 been identified. Larvae and juveniles move downstream into the low-salinity zone of the Delta

and Suisun Bay (Moyle et al. 1992; Dege and Brown 2004), where they remain through the summer and autumn as juveniles and pre-spawning adults (Feyrer et al. 2007; Nobriga and Feyrer 2008). They feed on zooplankton, mainly calanoid copepods (Moyle 2002). Longfin smelt generally spawn at the end of their second year, although some spawn in their third year (Moyle 2002; Rosenfield and Baxter 2007). Like delta smelt, longfin smelt spawn in fresh water, but in the late winter or early spring. Larvae and juvenile longfin smelt migrate downstream to low-salinity waters. Juveniles continue moving seaward, and most of the population matures in San Francisco Bay and the near-coastal ocean.

Striped bass is a long-lived species that was introduced to the Estuary in 1871 (Dill and Cordone 1997). The population grew rapidly and supported a commercial fishery (Moyle 2002). Despite declines in abundance, striped bass still support a recreational fishery. Striped bass typically first spawn at five years of age. Most spawning occurs in the lower Sacramento River in the spring. The semi-buoyant eggs and larvae move with the current into low-salinity reaches of the Delta (Dege and Brown 2004). Salinity and turbidity largely define the habitat of age-0 striped bass, which is confined mainly to the tidal portion of the Sacramento River upstream of its confluence with the San Joaquin River. Older striped bass occupy habitats throughout the rivers, estuary, and near-coastal ocean. Threadfin shad is a freshwater obligate that was introduced into Central Valley reservoirs as forage for largemouth bass (*Micropterus salmoides*), an introduced sport fish, and rapidly colonized downstream areas including the Delta (Dill and Cordone 1997). Threadfin shad are omnivores that can filter feed or sight feed on individual particles. They generally spawn at two years of age but may spawn at one or three years in the freshwater portions of the Delta during the late spring and summer (Moyle 2002). Juveniles and adults occur in the same freshwater areas.

193 In general, habitat for these fishes is defined on the basis of environmental conditions
194 rather than specific geographic locations. In wet years, for example, brackish water tends to
195 move seaward. In dry years, brackish water is more landward. Because the location of the
196 transition between brackish and fresh water varies as a function of precipitation and Delta
197 outflow, the extent of habitat can expand or contract depending on whether the transition is
198 located within the Delta, where channels tend to be narrow and deep, or in Suisun Bay, which is
199 wide and relatively shallow. Striped bass and threadfin shad are sensitive to low levels of
200 dissolved oxygen (Moyle 2002), which can occur during summer and autumn in the extreme
201 upstream limits of the south Delta (Lehman 2004) and in the some sloughs in Suisun Marsh
202 when they receive water from autumn drainage of reclaimed marshlands (Feyrer et al. 2007).
203 However, in the Delta as a whole, low dissolved oxygen is uncommon.
204 Overlap of all four species is limited (see Fig. 3). All occur in the freshwater portions of
205 the estuary in spring, and all but threadfin shad occur in the brackish water portion of the estuary
206 in spring. Rather than maturing to adulthood in the low salinity zone of the Estuary like delta
207 smelt and striped bass, juvenile longfin smelt migrate farther downstream to San Francisco Bay.
208 Thus, the spatial distribution of juveniles and adults varies among species. Delta smelt occur in
209 the Delta and Suisun Bay, longfin smelt occur in San Francisco Bay and the ocean, threadfin
210 shad are confined to the freshwater portions of the delta, and adult striped bass are found
211 throughout the Estuary.
212
213 Conceptual model of interactions between life stages and stressors
214

215 Fish and their foods may respond differently to sublethal exposures to contaminants depending
216 on life stage and how exposures occur. For example, the responses to pulsed pesticide inputs
217 after storms (Guo et al. 2007) can differ from the responses to chronic exposure to municipal
218 wastewater throughout the year (Dugdale et al. 2007). Whether pulsed or chronic, contaminant
219 exposure can cause organisms to divert energy from growth, reproduction, and predator
220 avoidance to mitigate sublethal contaminant effects (Beyers et al. 1999b; Goto and Wallace
221 2010). Vertebrates regulate sublethal effects physiologically by processes such as sequestration
222 (e.g. deposition of mercuric selenide as insoluble particles in the liver) (Yang et al. 2008),
223 incorporation of organic contaminants into fat reserves (Greenfield et al. 2008; Ostrach et al.
224 2008), enzymatic detoxification (Wallace et al. 2003), and excretion (Linton et al. 1998).
225 Organisms can also acclimate to chronic contaminant exposure via compensatory processes. For
226 example, many fish, including striped bass, can detoxify metals through greater production of the
227 metal-binding protein metallothionein (Geist et al. 2007; Spearow et al. 2011). Because
228 contaminants are agents of natural selection, over multiple generations some organisms develop
229 heritable resistance to aromatic contaminants such as polychlorinated biphenols (PCBs;
230 electronic coolants), polycyclic aromatic hydrocarbons (PAHs; by-products of combustion)
231 (Wirgin and Waldman 2004; Yuan et al. 2006; Wirgin et al. 2011), pesticides (Relyea and
232 Hoverman 2006), and metals (Meyer et al. 2003; Guinand et al. 2010). Compensatory processes
233 are not particularly efficient in developing fish. Depending on the species and the contaminant,
234 less than 10% of the dose lethal to an adult can be fatal to a juvenile (Boening 2000). Thus,
235 larvae and juveniles are the most sensitive life stages of pelagic fishes.

236 From late winter through early summer, just before and during spawning, several factors
237 may increase the vulnerability of delta smelt, longfin smelt, age-0 striped bass and threadfin shad

238 to toxic exposure (Fig. 3). During this period, the life histories, habitat, and geographic location
239 of the four fish species coincide when they migrate to the freshwater portions of the Delta to
240 spawn (Fig. 2). Moreover, spawning and the life stages that are most responsive to contaminants
241 overlap with periods of high rainfall. Winter and spring storms can increase nitrogen and
242 contaminant loading to concentrations that can be acutely toxic (Fig. 3, effect A) (Beighley et al.
243 2008; Kuivila and Hladik 2008). Storm runoff can double or triple mercury loads (Domagalski et
244 al. 2004) and increase concentrations of pesticides as much as 10-fold (Brady et al. 2006; Guo et
245 al. 2007).

246 Contaminants can kill fish prey, effectively mimicking resource limitation as can occur
247 from the introduction of a competitor (Rohr et al. 2006) (Fig. 3, effect F). This contaminant-
248 mediated food limitation occurs when contaminants are acutely toxic to invertebrates or algae,
249 but not to fish. For example, before the recent abrupt decline of fishes, agricultural effluents in
250 the upper Sacramento Basin apparently reduced survival of age-0 striped bass through toxicity to
251 their main prey, the opossum shrimp (*Neomysis mercedis*) (Bailey et al. 1994). Even in the
252 presence of pyrethroid pesticides (highly toxic to fish), direct mortality may have less of an
253 effect on abundance of fishes than pesticide-driven reduction in their prey (Fleeger et al. 2003;
254 Floyd et al. 2008). Moreover, food limitation can increase the susceptibility of organisms to
255 toxins. Without any change in their chemical environment, fishes and invertebrates die at lower
256 concentrations of toxins when starved than when fed *ad libitum* (Herbrandson et al. 2003;
257 Anderson et al. 2006; Ducrot et al. 2010). For example, in a paired laboratory and field study,
258 larval striped bass had low recruitment when chronic stress from contaminants (measured as
259 histologic pathologies) was paired with food limitation (Bennett et al. 1995). Fish growth
260 depends on adequate caloric intake but food quality is also important. Ng et al. (2009)

261 demonstrated that environmentally realistic concentrations of calcium in their foods protected
262 rainbow trout (*Oncorhynchus mykiss*) against both respiratory (branchial) and dietary uptake of
263 cadmium.

264 During egg production, female fish transfer contaminants to eggs, which can impair
265 development and survival of embryos (Fig. 3, effect E). For species with short life cycles, and
266 thus, limited time to accumulate toxins, adverse effects from maternal transfer to eggs are less
267 likely. Such species include delta smelt, which spawn at age 1 and then die, and threadfin shad
268 and longfin smelt, which spawn at age 2 or 3 and then die. However, striped bass begin spawning
269 at age 5 and survive to spawn multiple times (Moyle 2002); allowing more time to accumulate
270 persistent chemicals in their tissues that can be passed to their eggs. Chemicals found in bass
271 eggs include polychlorinated biphenyls, polybrominated diphenylethers (flame retardants), and
272 current-use and banned pesticides (Ostrach et al. 2008).

273 During migration to spawning grounds in the Delta, anadromous fish mobilize energy
274 reserves, which releases organic contaminants from adipose tissues (Debruyn et al. 2004) (Fig. 3,
275 effect M). Among the four pelagic fish considered here, increased exposure from release of
276 toxins from fat reserves is probably greatest for the anadromous striped bass and longfin smelt,
277 which travel to the Delta from San Francisco Bay and the coastal ocean (Moyle 2002). Release
278 of contaminants from fat is probably least for threadfin shad, which are confined to the
279 freshwater portions of the Delta and upstream watersheds (Dill and Cordone 1997).

280 The survival and health of fishes can be strongly affected by pollutants that alter behavior
281 and immune function (Eder et al. 2007) (Fig. 3, effect I). Sublethal exposure of larvae to
282 contaminants can impair swimming performance and increase vulnerability to predators (Floyd
283 et al. 2008). Common environmental concentrations of copper, organophosphates, and

284 pyrethroids have been shown to impair salmonid olfaction (Sandahl et al. 2007). Sense of smell
285 is important for these migratory animals to find food and locate their natal stream. Olfactory
286 damage of juveniles leaving their natal stream prevents them from imprinting properly, making it
287 impossible for them to return to the correct stream for spawning (Sandahl et al. 2007). Regarding
288 immunological effects, statistically significant alterations in gene expression of factors involved
289 in neurological immune function occurred in delta smelt after their 24-h exposure to a low
290 concentration (0.0625 $\mu g\ L^{-1}$) of the pyrethroid pesticide, esfenvalerate (Connon et al. 2009).
291 Pelagic fishes have different life histories and use different regions of the Delta and
292 Estuary during different seasons; as a result, contaminant exposure also tends to vary among
293 locations and seasons. For example, central California's climate is conducive to extensive
294 blooms of *Microcystis aeruginosa*, a toxic cyanobacterium that became abundant around 1999 in
295 the warm, fresh waters of the central and southern Delta (Fig. 3, effect H). Depending on flow
296 and temperature, blooms can extend westward into the low salinity zone (Fig. 1). Of the four
297 pelagic fishes, only longfin smelt likely do not encounter the blooms because their population
298 mostly occurs downstream of Suisun Bay during the summer bloom period (Fig. 3).
299 Some environmental conditions can increase the susceptibility of organisms to
300 contaminants (Fig. 3, all seasons). Salmonids exposed to brackish salinities increase their output
301 and activity of the enzymes (flavin containing monooxygenases) that convert fenthion, an
302 organophosphate biocide, into more toxic forms. Thus, salinity increases the toxicity of fenthion
303 (Lavado et al. 2009a). Salinity also requires energy for osmoregulation. Osmoregulatory stress
304 can alter metabolic rate and increase susceptibility of fish to copper (Pistole et al. 2008). Costs of
305 resistance to one contaminant can increase susceptibility to another stressor or decrease
306 fecundity. For example, killifish (*Fundulus heteroclitus*) survived exposure to sediment

307 contaminants from the Elizabeth River in Virginia (U.S.A.), but the fish died when subjected to
308 the same contaminant concentrations combined with the additional stressors of photoenhanced
309 toxicity and hypoxia (Meyer and Di Giuliuo 2003). Fish adapted to elevated cadmium exhibit
310 reduced fecundity, reduced offspring size, and longer maturation time, suggesting energetic
311 trade-offs in resource allocation (Xie and Klerks 2004).

312 Turbidity and temperature can alter susceptibility of fishes to contaminants. Increased
313 turbidity can limit penetration of ultraviolet radiation, which protects organisms from
314 photoactivation of PAHs (Ireland et al. 1996). However, greater turbidity can decrease the
315 energy intake of visual predators by hampering their feeding success. Decreased turbidity can
316 also limit the food intake of larval fish by increasing their perceived threat from predators,
317 causing them to hide rather than forage (Domenici et al. 2007). Regarding temperature (Fig. 3,
318 effect T), when juvenile and adult rainbow trout were exposed to temperatures approaching their
319 thermal maximum, sublethal concentrations of ammonia significantly impaired protein
320 metabolism and diminished growth in starved fish but also in fish fed *ad libitum* (Linton et al.
321 1997; Morgan et al. 2001).

322

323 Contaminant stressors

324

325 The Delta receives many natural and anthropogenic compounds from agricultural and urban
326 areas. Although agricultural applications of pesticides are well documented, transport
327 mechanisms for most pesticides from fields to waterways are not well understood and
328 concentrations of most pesticides are not monitored (Kuivila and Hladik 2008). Urban
329 applications are not well documented, nor is transport well characterized. Thus, the transport,

330 fate, and ultimate loads of all chemicals entering the Delta cannot be rigorously quantified and
331 there are no long term monitoring programs that document concentrations (Johnson et al. 2010).
332 Accordingly, we used available data from the Estuary to qualitatively evaluate the biological
333 importance to fish of four categories of major contaminants — metals, nutrients, toxic algae, and
334 pesticides. These contaminants are common to populated estuaries worldwide (Matthiessen and
335 Law 2002; Ducrotoy et al. 2000; Luo et al. 2004; Bollmohr et al. 2007; Couillard et al. 2008).
336 We also briefly review other contaminants that may have been chronic contributors to the long-
337 term decrease in fish abundance but probably did not cause the recent declines.
338
339 Metals and the metalloid selenium
340
341 There is no evidence that acute toxicity of selenium, mercury, or copper contributed to recent
342 declines of pelagic fishes in the Estuary, but chronic effects may have played a role in the long-
343 term declines. The metalloid selenium is one of the most difficult elements to regulate because it
344 is an essential trace nutrient that becomes a potent toxicant if consumed in excess of nutritional
345 requirements. Fish uptake of selenium varies as a function of diet with greatest uptake and toxic
346 effects on benthic feeding top predators (Stewart et al. 2004). Selenium exposure during
347 development causes diagnostic skeletal deformities that affect swimming performance (Chapman
348 et al. 2010). At this time, it appears that neither irrigation of seleniferous soils in the San Joaquin
349 River drainage nor selenium from industrial sources along Suisun Bay (Doblin et al. 2006) affect
350 pelagic fishes; although it was a concern in the past (Luoma and Presser 2009).
351 Mercury in the Estuary is derived from geothermal features and from mining in the Delta
352 watershed during the mid 1800s (Domagalski et al. 2004). Urban runoff and wastewater also

353 contribute mercury, copper, silver, and other metals (Flegal et al. 2005). Concentrations of
354 inorganic mercury in the Delta range from undetectable to 2 μg L^{-1} (Johnson et al. 2010).
355 Sublethal mercury concentrations between 1 and 50 μg L^{-1} can reduce fertility, slow growth, and
356 impair neurologic and endocrine functions of fish (Boening 2000).

357 Selenium and mercury accumulation are positively correlated in the tissues of many
358 organisms. Observations of coinciding and ostensibly lethal concentrations of mercury and
359 selenium in wild marine organisms led to the hypothesis that selenium protected organisms
360 against mercury toxicity (Koeman et al. 1973; Dietz et al. 2000) The protective effect of
361 selenium against mercury has been supported by laboratory studies of model vertebrates and cell
362 cultures (Yang et al. 2008). Nonetheless, protection varies depending on the form of both
363 selenium (selenite, selenate, selenocysteine, selenomethionine) and mercury (total mercury,
364 inorganic mercury, methylmercury), routes of administration for each (dietary or respiratory),
365 and whether organisms are dosed with selenium before, during, or after mercury exposure (Yang
366 et al. 2008). An example of the complexity of dose and uptake comes from seleniferous lakes in
367 Canada where concentrations of selenium and mercury in the tissues of young walleye
368 (*Stizosedion vitreum*) are negatively correlated even though mercury exposure from upwind
369 smelting operations is high. These findings suggest that chronic exposure to high selenium
370 concentrations may prevent mercury uptake (Yang et al. 2010). It appears that eventual
371 formation of inert, inorganic mercuric selenide depends on initial formation of selenoproteins
372 (selenomethionine and selenocysteine) as well as the availability of the antioxidant enzyme
373 glutathione, which mediates the process (Yang et al. 2008; Yang et al. 2011). In fish that are
374 native to the Estuary, the protective effect of selenium against methylmercury toxicity was
375 demonstrated in laboratory studies on larval Sacramento splittail (*Pogonichthys macrolepidotus*)

376 that were fed selenomethionine before being fed a dose of mercury (Deng et al. 2008).
377 Laboratory findings have not been replicated in the field or laboratory for other species. Fishes in
378 the Estuary contain mercury and selenium but tissue concentrations of these elements were not
379 reported as ratios within the same individual, and thus cannot be related to possible protection
380 against mercury toxicity (Saiki et al. 1992; Greenfield et al. 2005).

381 Before remediation, Iron Mountain Mine in the Sacramento River watershed discharged
382 substantial amounts of metals including copper, with toxic effects on anadromous salmonids
383 returning to natal streams; however, the mine has met remediation criteria since 2003 (USEPA
384 2008). Current copper loadings from agricultural and urban runoff reflect the common use of
385 copper sulfate as an algaecide. Purchases of copper sulfate in Delta counties totaled 1.4 to 1.8
386 million kg/yr between 1996 and 2004 (CDPR 2009). Copper is highly toxic to invertebrates and
387 larval fish, damaging DNA and impairing immune function and behavior. Copper concentrations
388 as high as 4,032 $\mu g\ L^{-1}$ were reported between 2004 and 2007 (Johnson et al. 2010), although
389 surface water concentrations in the Delta typically average 2 $\mu g\ Cu\ L^{-1}$ (P. Lehman, unpubl.
390 data). The latter concentration can halve the swimming performance of salmon in freshwater
391 (Sandahl et al. 2007). Juvenile delta smelt are relatively sensitive to copper, with a median lethal
392 concentration of 17.8 $\mu g\ L^{-1}$ after a 7-d exposure. Moreover, patterns of gene expression
393 indicated likely impairment of swimming performance, digestion, and immune function (Werner
394 et al. 2009; Connon et al. 2011). Complexation of copper with inorganic anions (mainly
395 carbonates) and natural organic carbon reduces the available fraction of copper (Brooks et al.
396 2007a; Brooks et al. 2007). Estimates of the biologically available fraction indicate that copper is
397 not acutely toxic to fish at the environmental concentrations found in San Francisco Bay (Buck

et al. 2007). However, neither acute nor sublethal effects of mercury or copper have been directly tested in waters from the Bay.

Aluminum is not a major toxicant in the alkaline waters of the Estuary (pH range 7.2 to 8.0) (USGS 2011). Aluminum toxicity occurs in acidified waters in parts of Puget Sound (Washington, U.S.A.) (Landis et al. 2004), the southern regions of the Canadian Shield, and the northeastern United States, including the upper tributaries of Chesapeake Bay (Maryland and Virginia) where granitic soils have little buffering capacity (Hall et al. 1993). In acidified waters below pH 6.0, aluminum occurs as monomeric species (Al^{3+}, $AlOH^{2+}$, $Al(OH)_2^{+}$, $Al(OH)_3°$, and $Al(OH)_4^{-}$) that deposit on gill surfaces and smother the animal. However, when pH remains above 6.0, monomeric aluminum binds with carbonates and the toxic forms of aluminum are absent (Gensemer and Playle 1999). Moreover, aluminum is not toxic when bound into particles by iron, natural organic matter (Roy and Campbell 1997), or silica. However, under acidic conditions found in the upper reaches of some estuaries, aluminum is highly toxic to larval striped bass and salmonids (Hall et al. 1993; Nilsen et al. 2010). Even after anadromous salmon migrate to waters with a neutral pH, transient exposure to aluminum in mildly acidic water significantly decreases survivorship and growth (Kroglund et al. 2007).

Inorganic nitrogen

Of the three forms of nitrogen commonly used in fertilizers – ammonia (NH_3), ammonium (NH_4^{+}), and nitrate (NO_3^{-}) – only NH_3 is acutely toxic to fish. Agricultural inputs of commercial fertilizer to Delta waterways did not increase after 2000, but the human population within 30 km of the Delta grew by about 11% (350,000) between 2000 and 2007. Nitrogenous input into

421 waterways from wastewater treatment plants increases by ~1 metric ton/d per 100,000 people
422 (Jassby 2008). Primary treatment (solids removal) and secondary treatment (microbial
423 consumption of organics) of water is sometimes followed by advanced-secondary treatment that
424 removes NH_3 and NH_4^+ but not NO_3^-. Except for the city of Stockton, which initiated advanced-
425 secondary water treatment in 2004, major cities in the Delta region, such as Sacramento, do not
426 conduct this final water treatment (Table 1). Whether from fertilizers or wastewater plants,
427 nutrient concentrations as high as 941 μmol NH_3 L^{-1} and 7,800 μmol NO_3^- L^{-1} (Johnson et al.
428 2010) have made some Delta waters unsuitable for human use and aquatic organisms (Gowdy
429 and Grober 2003). (Criteria for drinking water are 1,800 μmol NH_3 L^{-1} and 161 μmol NO_3^- L^{-1};
430 criteria for freshwater organisms are 176 to 300 μmol NH_3 L^{-1} depending on pH, temperature,
431 and life stage).

432 High nutrient loadings can have complex repercussions in waterways. For example, NH_4^+
433 may indirectly limit availability of food for pelagic fish (Fig. 3, effect F). Although NH_4^+ is a
434 plant nutrient, some primary producers use NO_3^- more efficiently. Moreover, elevated NH_4^+ (> 4
435 μmol L^{-1}) inhibits NO_3^- uptake by phytoplankton, including diatoms. Lower diatom production
436 directly reduces abundances of this large-diameter algal species that fall within the optimum size
437 range for adult copepod food and are positively correlated with zooplankton biomass in the Delta
438 (Lehman 2004; Dugdale et al. 2007). Currently, high NH_4^+ concentrations in Suisun Bay prevent
439 diatom blooms in spring (Dugdale et al., 2007). Diatom blooms are believed to provide optimal
440 conditions for production of zooplankton during a critical period in the nutrition and growth of
441 several pelagic fishes, including delta smelt.

442 Throughout the Delta, sublethal concentrations of NH_4^+ and NH_3 were negatively
443 correlated with 10-day growth of the amphipod *Hyalella azteca. H*owever, concentrations that

444 cause effects in *Hyalella* are far higher than those found in the water samples. Historical data
445 from the Delta suggest that increasing NH_4^+ loads since 1982 might be associated with output
446 from sewage treatment plants (Jassby 2008). Possibly toxicity was due to a mixture of ammonia
447 with other contaminants or other contaminants in which ammonia happened to co-occur
448 (Werner et al. 2010), suggesting that ammonia could act as an indicator of water quality.
449 Nutrients can affect oxygen availability to ectotherms during summer and autumn when
450 flows are minimal and warmer waters diminish the pool of available oxygen, which intensifies
451 any hypoxia caused by nocturnal respiration of algal blooms. Although high nitrogen levels
452 associated with the Stockton wastewater treatment facility once triggered blooms of primary
453 producers that increased oxygen demand and promoted hypoxic conditions (Gowdy and Grober
454 2003; Lehman 2004), new facilities with advanced secondary treatment (Table 1) have
455 eliminated this problem. Currently, low dissolved oxygen rarely occurs in the Delta due to low
456 light and high vertical mixing. Hypoxia might occur in some smaller sloughs with poor
457 circulation but, to our knowledge, it has been documented only for Suisun Marsh. In general, the
458 waters of the Delta where pelagic fish live are not hypoxic.
459
460 Cyanobacterial blooms
461
462 Although blooms of cyanobacteria such as *M. aeruginosa* are not usually classified as
463 contaminants, they can be directly toxic to fish (Fig. 3, effect A) or reduce the quality or quantity
464 of food (Fig. 3, effect F). In the Delta, blooms of *M. aeruginosa* have recurred each summer
465 since 1999 (Lehman et al. 2008) (Fig. 1) except for summer 2010, which had high flow and
466 unseasonably cool water temperatures in August and September (Lehman, unpubl. data). *M.*

467 *aeruginosa* produces several forms of the toxin microcystin, which cause liver cancer in humans
468 and fish (Ibelings and Chorus 2007; Ibelings and Havens 2008). Juvenile striped bass collected
469 from areas where *M. aeruginosa* was abundant in the Delta contained microcystins and
470 expressed liver biomarkers consistent with carcinogen exposure (Figure 3, effect H) (Lehman et
471 al. 2010). Long-term exposure to diets containing microcystins reduces survival of embryos and
472 inhibits the growth, impairs liver function, and decreases the RNA:DNA ratio of juveniles and
473 adults (Malbrouck and Kestemont 2006; Deng et al. 2010) in Medaka *(Oryzias latipes)* and
474 native splittail. Blooms are most likely to affect threadfin shad and age-0 striped bass, species
475 that remain in the freshwater portions of the Delta in summer, but high flows move these toxic
476 cyanobacteria into brackish portions of the estuary where other fish, including delta smelt, may
477 be affected (Lehman et al. 2005). Delta smelt could be affected in freshwater tidal habitats
478 upstream or downstream in brackish water.

479 *M. aeruginosa* may also produce cascading effects on the food web (Fig. 3, effect F).
480 Allelopathy (biochemicals that influence the growth, survival, and reproduction of other
481 organisms) associated with cyanobacterial blooms can affect the base of the food web by altering
482 phytoplankton community composition (Suikkanen et al. 2005). Because *M. aeruginosa* tolerates
483 high temperatures and nutrients, they can further alter community composition by outcompeting
484 other algal and macrophyte species for light and trace nutrients (Munkes 2005). The larvae of all
485 four pelagic fishes considered here feed on small zooplankton. As delta smelt mature they
486 mainly consume calanoid copepods, whereas threadfin shad shift to an omnivorous diet of
487 zooplankton and phytoplankton. Juvenile longfin smelt and striped bass shift to feeding on
488 copepods and opossum shrimp. Later, juvenile bass feed on small fish. Whereas cyanobacteria
489 are generally less nutritious than other phytoplankton, laboratory and field results suggest that

490 zooplankton can feed profitably (greater energy intake than energy cost) on cyanobacteria
491 (Wilson et al. 2006; Tillmanns et al. 2008). Shifts in the composition of the zooplankton
492 community indicate that some zooplankton feed more profitably on cyanobacteria than others
493 (Lehman et al. 2010). In the Delta, fewer diatoms and green algae and more cryptophytes
494 occurred where *M. aeruginosa* was abundant (Lehman et al. 2010). Because diatoms and green
495 algae have the optimum feeding diameter for zooplankton, their loss limits the availability of
496 phytoplankton food with high carbon content, particularly in the low salinity zone where pelagic
497 fish are abundant (Lehman 2000). The co-occurrence of other cyanobacteria with *M. aeruginosa*
498 may further decrease the quality of zooplankton food during cyanobacterial blooms.

499 Whether *M. aeruginosa* blooms directly affect zooplankton in the Delta is less clear.
500 Calanoid copepods can selectively avoid feeding on *M. aeruginosa* and population sizes were not
501 lower within cyanobacterial blooms, but the presence of microcystins in their tissue and the
502 adverse affect of this toxin on health and survival of the critical zooplankton prey species in the
503 Delta, the copepods *Pseudodiaptomus forbesi* and *Eurytemora affinis*, in laboratory tests
504 suggests populations may not reach maximum sizes (Ger et al. 2009; Lehman et al. 2010). *M.*
505 *aeruginosa* blooms in the Delta were associated with a decreased ratio of cladocerans to calanoid
506 copepods, which suggests a reduced relative abundance of large cladocerans for consumption by
507 predators. To our knowledge a cascading effect from cyanobacteria to zooplankton to
508 zooplankton predators has not been investigated in natural waters.
509
510 Pesticides in current use
511

512 Agricultural, commercial, and urban purchases of pesticides within the Delta and the upstream
513 watershed averaged 21 million kg annually from 1990 to 2007, the period of record (CDPR
514 2009). Urban applications are not monitored. Agencies test concentrations in waterways of about
515 half of 161 pesticides applied in annual amounts greater than 500 kg by agricultural and
516 commercial users. Those that are monitored are detected in runoff, mainly from December
517 through July (Kuivila and Hladik 2008), which is the period of fish spawning and larval
518 development (Fig. 3, effect A). Organophosphates such as diazinon and chlorpyrifos have
519 repeatedly exceeded water quality standards, and contributed to listing of the Delta as an
520 impaired waterway. Carbamate pesticides (carbaryl, carbofuran) and organophosphates from
521 agricultural and urban drainages explained 23 to 63% of the variation in recruitment of larval
522 striped bass in 1989–1991 (Bailey et al. 1994). In 1994 and 1995, organophosphate and
523 carbamate insecticides in water samples collected throughout the Delta intermittently caused
524 acute toxicity to *Ceriodaphnia dubia*, an invertebrate surrogate for Delta prey species (Werner et
525 al. 2000). Concerns about the toxicity of organophosphates to humans resulted in reduced use
526 (primarily of diazinon) by 40% to 79% as of 2000 (Dileanis et al. 2002; Kratzer et al. 2002).
527 Nonetheless, the most commonly applied insecticide in 2006 was chlorpyrifos (198,495 kg)
528 (Kuivila and Hladik 2008), an organophosphate that is toxic to zooplankton and fish (Eder et al.
529 2007). Pyrethroid pesticides, the favored replacement for organophosphates, are highly toxic to
530 fishes and a million times more toxic than organophosphates to invertebrates (e.g. *Eurytemora*
531 *affinis*), causing paralysis and eventual death (Werner and Moran 2008). The trend toward
532 greater pyrethroid use has coincided with abrupt declines in abundances of pelagic fishes.
533 The source of pyrethroids appears to be largely urban. Seventy-five percent of the
534 198,295 kg of pyrethroids used in 2006 (Kuivila and Hladik 2008) were applied in urban areas

535 (Spurlock and Lee 2008). During 2008 and 2009, Weston and Lydy (2010) investigated the
536 toxicity of pyrethroids by exposing *H. azteca* to water samples from point sources (3 wastewater
537 treatment plants) and non-point sources (6 urban and 8 agricultural locations). About one-third of
538 samples from agricultural runoff contained pyrethroids, and 10% of samples contained toxic
539 concentrations. In contrast, nearly all samples of non-point urban runoff from Sacramento,
540 Stockton, and Vacaville were toxic. Those samples contained pyrethroid concentrations of 8 to
541 20 ng L^{-1}, which are 4 to 10 times the concentration that causes paralysis in *H. azteca* (Weston
542 and Lydy 2010).

543 In municipal wastewaters, advanced-secondary treatment appears to mitigate the toxicity
544 of pyrethroids. The city and county of Sacramento are served by the County of Sacramento
545 Regional Wastewater Treatment Plant, which does not conduct advanced-secondary treatment.
546 This plant discharged pyrethroids at concentrations higher than at any other location, and every
547 sample tested was toxic to *H. azteca* (Weston and Lydy 2010). One third of samples collected in
548 outflow from the Vacaville plant were toxic. At the Stockton plant, which conducts advanced-
549 secondary treatment, pyrethroids were infrequently detected and toxicity was never observed.
550 Regardless of source, at 16 mainstem sites across the Delta, pyrethroids in conjunction with
551 ammonia and potentially organophosphates impaired growth, or survival of *H. azteca* in about
552 10% of 701 water samples collected between January 2006 and December 2007 (Werner et al.
553 2010).
554
555 Other contaminants
556

557 Several categories of contaminants may be having long-term effects on the condition of fish in
558 the Delta. Applications of several pesticides have not increased or have ceased in recent years.
559 Consequently, they are unlikely to drive recent declines in abundance of fishes, although as
560 chronic stressors they may have contributed to longer-term declines. We discuss them given
561 relatively high concern, yet limited knowledge, about their potential effects. Organochlorine
562 insecticides, mainly dichlorodiphenyltrichloroethane (DDT; banned in 1972 in the United States)
563 and DDT breakdown products, are potent neurotoxins. Because these persistent chemicals bind
564 strongly to sediments they probably have minimal effect on pelagic fishes, except during
565 sediment resuspension by dredging operations or if fish consume benthic organisms.
566 Nonetheless, DDT and breakdown products can accumulate in long-lived species such as striped
567 bass (Ostrach et al. 2008).

568 Urban and industrial sources are the main contributors of PCBs and PAHs. The PCBs,
569 banned since 1976, suppress immune response in aquatic biota and humans (Ostrach et al. 2008;
570 Ankley et al. 2009). Most adult fish can enzymatically neutralize PAHs, but these carcinogens
571 are highly toxic to early life stages. Nearly all sediments sampled in the Delta contained PCBs
572 (88%) and PAHs (98%) in 2006. Many sediment samples (68%) exceeded 2.5 μg PCB kg^{-1}
573 organic carbon, which is equivalent to 2.5 times the U.S. Environmental Protection Agency's
574 criterion for freshwater organisms (SFEI 2007).

575 Many compounds, from flame retardants and plasticizers to pharmaceuticals and personal
576 care products, remain in municipal wastewater after treatment (Ankley et al. 2009), particularly
577 during the summer when rainfall is virtually nil in the Sacramento-San Joaquin Valley (Loraine
578 and Pettigrove 2006). Long-lived fish such as striped bass can transfer PCBs, polybrominated
579 diphenyl ethers (PBDEs, flame retardants), and chlorinated pesticides to their eggs (Fig. 3, effect

580 E), producing larvae with abnormal brain and liver development and impaired growth compared
581 to hatchery-reared striped bass (Ostrach et al. 2008). Concentrations of flame retardants are high
582 in clams (*Corbicula fluminea*) from the Sacramento and San Joaquin rivers. Concentrations of
583 PBDE in tissues of striped bass increased 4-fold, from ~400 to ~2000 pg g^{-1} lipid, between 1997
584 and 2003 (Hoenicke et al. 2007). Many of these compounds, which mimic natural endocrine
585 hormones, can affect reproduction of wild fish (Kidd et al. 2007; Lavado et al. 2009b). However,
586 few endocrine disruptors are monitored in the Estuary (Hoenicke et al. 2007). When monitored
587 in agricultural drainages, natural estrogens from human wastewater and synthetic estrogens from
588 oral contraceptives were seldom above detection limits (Lavado et al. 2009b).

589 Unexpected interactions can stem from pharmaceutical inputs. For example, Robinson et
590 al. (2005) exposed 5 aquatic organisms (a cyanobacterium, *M. aeruginosa*; duckweed, *Lemna*
591 *minor*; a green alga, *Pseudokircheriella subcapitata*; a crustacean, *D. magna*; and a fish, *P.*
592 *promelas*) in the laboratory to fluoraquinolone antibiotics at concentrations that occur in the
593 field. Of the organisms tested, *M. aeruginosa* was most sensitive to the antibiotic (Robinson et
594 al. 2005). Because 70 to 90% of antibiotics are excreted unchanged (Lopes de Souza et al. 2009;
595 Kim et al. 2011), wastewater from hospital and veterinary use might inhibit bacterial growth that
596 could have either positive or negative effects on fishes. Input can be high (Lopes de Souza et al.
597 2009), but a fate and transport study showed that 90% of the antibiotics chlortetracycline,
598 sulfamethazine, and tyolsin were removed by abiotic processes such as adsorption to soil
599 particles (Kim et al. 2011). To our knowledge, antibiotics are not present in Delta waters,
600 although wastewater effluents in the Delta contain insecticides, herbicides, plasticizers, perfume
601 and lotion stabilizers, hypertension medication, and analgesics (Loraine and Pettigrove 2006).
602

603 Potential effects of climate change

604

605 Climate models project increased frequency of severe storms, extended drought, and 2 to 6ºC
606 increases in air temperatures in California by 2100 (Cayan et al. 2006). Aside from potential
607 effects on salinity and turbidity that greatly influence the habitat quality of fishes (Feyrer et al.
608 2010), such patterns could increase the frequency of pollutant pulses from overland runoff during
609 storms, increase the frequency and severity of droughts that might concentrate contaminants
610 through evaporation, and raise water temperatures that increase metabolic rates of ectotherms
611 and alter their susceptibility to contaminants. For example, storms can increase mercury and
612 diazinon concentrations at sites within the San Francisco watershed 10 to 100-fold, exceeding
613 water quality criteria (Domagalski et al. 2004; Guo et al. 2007). Reduced summer flows (i.e., less
614 dilution) and evaporation may increase instream contaminant concentrations. However, drought
615 may also decrease the flushing of solutes into waterways.

616 Without any change in water chemistry, warming water temperatures can convert
617 sublethal concentrations to biologically lethal levels or *vice versa.* In ectotherms, warming water
618 temperatures can increase contaminant dose because their respiration and feeding rates must
619 increase to support higher metabolic rates. However, the response to higher dose varies
620 depending on whether enzymatic breakdown or elimination can keep pace with uptake and
621 whether contaminants are converted physiologically to more toxic forms. The effect of higher
622 metabolic rate on biologic response also depends on the mode of action of the contaminant (e.g.
623 respiratory failure versus internal bleeding), and whether the relationship between temperature
624 and toxicity is linear or exponential (Harwood et al. 2009).

For example, most pyrethroids (Weston et al. 2009) and DDT have greater toxicity at lower temperatures. Some pyrethroids are exponentially more toxic at lower temperatures because enzymatic breakdown slows. Toxicity of pyrethroids appears unchanged at temperatures above about 23°C (Weston et al. 2009), probably because the efficiency of enzymatic processes does not improve above a thermal optimum. Physical effects of DDT, also a neurotoxin, increase at lower temperatures because nerve sensitivity increases (Harwood et al. 2009). Conversely, raising water temperature by 1 to 2°C caused 1.5- to 100-fold increases in the toxicity of metals, carbamates, organophosphates (including chlorpyrifos), and some organochlorines other than DDT (Lydy et al. 1990; Boening 2000; Heugens et al. 2001; Harwood et al. 2009). Large increases in susceptibility to toxins can occur when metabolic rates elevated by warmer temperatures increase uptake of pollutants. The higher dose apparently overwhelms excretion and enzymatic depuration. For compounds that are physiologically converted to more toxic forms (e.g. chlorpyrifos) the higher rate of metabolic transformation also increases their toxicity (Harwood et al. 2009).

Warming water temperatures can indirectly and directly reduce or tax the energy reserves that fish require to mitigate pollutants. Warmer water indirectly reduces caloric intake because fish become lethargic and decrease their foraging activity, which lowers consumption. Because warmer waters carry less oxygen (e.g. oxygen saturation in freshwater decreases by ~8% as temperature increases from 20 to 25°C), higher water temperatures can trigger the first mechanism of thermal intolerance—failure to meet oxygen demand because oxygen supply is inadequate. For example, some fish species require cool night temperatures to offset thermal stress during the day (Caissie 2006; Portner and Knust 2007), indicating that diminished nocturnal oxygen constitutes a hypoxic environment for fish. Fish make several physiologic

adjustments to tolerate hypoxia, initially increasing their basal metabolic rate, which directly depletes their energy reserves. Eventually, their survival depends on a shift to time-limited anaerobic metabolism that causes the buildup of organic acids (e.g. succinate) and reactive oxygen species in tissues, increasing the demand for antioxidants (Portner and Knust 2007). Food limitation may speed the pace and severity of this process because energy reserves are inadequate for diverting blood supply to critical functions and for meeting greater antioxidant demand (Dupont-Prinet et al. 2009).

While hypoxia is not currently a general concern in the Estuary, models indicate that future water temperatures in the Delta will exceed the thermal tolerance of delta smelt for much of the year (Wagner et al. 2011). Effects of climate change will vary among species depending on trade-offs between decreased toxicity of some pesticides and greater susceptibility to other contaminants, the timing and magnitude of chemical uses, hydrologic control of exposure spikes, and predator-prey interactions. Ultimately, higher minimum daily water temperatures (Fig. 4) could be a predictor of the toxic susceptibility of contaminant mixtures and energetic costs on pelagic fishes.

Emerging research approaches

In future research, use of energy as a common currency may increase the ability to evaluate contaminant interactions with oxygen demand, growth, reproduction, and ultimately survival of wild populations. For example, molecular biomarkers such as thyroid hormone levels or gene expression (Baker et al. 2009; Brar et al. 2010), when coupled with histological assays of body condition, can reveal the energetic consequences of suppressed thyroid function and tumor

671 growth in wild-caught organisms. Field studies are important in determining how energetic
672 responses to contaminants are mediated by diel temperature variability, seasonal hydrology,
673 resource availability, and predator-prey relationships (Clark and Clements 2006; Buchwalter et
674 al. 2007). From such data, researchers are developing bioenergetics models that predict
675 contaminant effects on individual species in conjunction with non-contaminant stressors (Beyers
676 et al. 1999a; Beyers et al. 1999b; Nisbet et al. 2000). For example, a bioenergetics model
677 explained why 40- to 50-fold higher concentrations of metals were required to cause equivalent
678 mortality in well-fed macroinvertebrates held at constant temperature in the laboratory compared
679 to the same species in the field (Buchwalter et al. 2007). Moreover, energetics are key to
680 predicting how ectotherms will respond to chronic contaminant exposure under changing water
681 temperatures and flow regimes.

682 Progress has been made toward developing models of multiple stressors that predict
683 change over time. Thomson et al. (2010) used Bayesian change-point analyses to examine the
684 potential effect of abiotic and biotic factors on the decline of the 4 fish species discussed in this
685 paper. Covariates included many of the non-contaminants that we discuss herein (e.g.
686 temperature increases, reduction or variation in food) but did not include contaminants; however,
687 after about 2000 those covariates no longer explained considerable variation in abundances.
688 Dynamic energy budget theory evaluates effects of multiple stressors on growth and
689 development in single species (Fujiwara et al. 2005; Jager and Klok 2010), and recently has been
690 applied to explore toxic effects of *Microcystis* on a cladoceran under controlled laboratory
691 conditions (Billoir et al. 2009). Dynamic energy budget theory has been used to assess causes of
692 apoptosis (programmed cell death), endocrine disruption, and ovarian tumors in the longjaw
693 mudsucker (*Gillichthys mirabilis*) in salt marshes on the Pacific coast of the United States

694 (Anderson et al. 2006). These approaches hold promise for quantifying effects of individual
695 stressors at one point in time, but a shortcoming of dynamic energy budget theory is that it
696 cannot perform life cycle assessments (Nisbet et al. 2010). Life cycle analyses evaluate
697 susceptibility versus resilience during changes in habitat and feeding at different life stages, but
698 the data needed for building such models for multiple, interacting species subjected to multiple
699 stressors are limited. Significant benefits could result from consistent, long-term monitoring of
700 water temperature, geochemistry, and contaminants, and fish abundances, age-size ratios and age
701 classes (as potential proxies for energetic costs and reproductive success). When laboratory
702 experiments are paired with acute and sublethal responses to contaminant mixtures in field
703 exposures, we contend that energetics approaches will make it possible to discriminate sublethal
704 effects of contaminants from those of other stressors causing declines of pelagic fishes in
705 systems such as San Francisco Estuary. Field exposures within cages or diversion structures can
706 provide opportunities to evaluate growth and biomarkers under environmentally realistic
707 conditions (Hall et al. 1993; Oikari 2006; Ings et al. 2011). As such data become available,
708 mensurative comparisons of field data in different locations (i.e. varied conditions) with models
709 of energy budgets will clarify the net effects of multiple sublethal factors on fish abundances.
710
711 Conclusions
712
713 Researchers from around the world, including Europe (Matthiessen and Law 2002; Daufresne et
714 al. 2007), South Africa (Bollmohr et al. 2007), and Asia (Luo et al. 2004), are struggling to
715 quantify relationships between altered water quality, non-contaminant stressors, and declines of
716 aquatic organisms. Although the San Francisco Estuary is one of the best studied estuaries in the

717 world, the ecological effects of contaminants remain unquantified, and are difficult to investigate
718 with standard methods based on acute toxicity. Our review of available evidence suggests that
719 although sublethal contaminant effects are not the sole cause of past and ongoing fish declines,
720 chronic bioenergetic costs resulting from exposure to metals, nitrogen-rich effluents,
721 cyanobacterial blooms, and pesticides are likely contributors.

722 We have shown that accounting for seasonal and geographic patterns of life histories can
723 clarify mechanisms important to particular life stages among species with habitats that range
724 from anadromous to brackish to freshwater (Fig. 3). The greatest threat to pelagic species most
725 likely occurs during larval life stages in the freshwater reaches of the Delta in late winter and
726 spring. During this period, storms may greatly increase concentrations of toxicants when the
727 physiologic ability of larvae to cope is undeveloped, when food limitation can result from acute
728 toxicity of runoff to zooplankton prey, when competition from invasive species for food occurs,
729 and when mixtures of contaminants are directly toxic or may simply have sublethal, inhibitory
730 effects.

731 In general, sublethal, chronic exposure to contaminants may reduce reproduction,
732 suppress immune function, increase expression of stress biomarkers (Fleeger et al. 2003), and
733 damage olfactory capability (Sandahl et al. 2007). Sublethal contaminant exposure can impair
734 immune function and swimming ability of delta smelt (Connon et al. 2011) and alter stress
735 response and reproductive success in striped bass (Geist et al. 2007; Ostrach et al. 2008). We
736 conclude that predicting the response of different fish species to contaminants requires
737 considering the sensitivity and exposure of different life stages, the energy deficits due to
738 multiple stressors, and the joint effects of temperature on metabolic rate and contaminant
739 elimination.

740

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746

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748

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1190 **Table**

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1193 Table 1 Wastewater effluent outputs in cities in the Sacramento and San Joaquin valleys

1194 (California, U.S.A.) as of 2006. Secondary treatment removes organic molecules, and advanced-

1195 secondary treatment removes NH_3, NH_4^+, and NO_3^-. Data source:

1196 http://www.water.ca.gov/publications/browse.cfm?display=topic&pub=120,382,10872

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Location	Final Step in Wastewater Treatment		Total Wastewater Volume
	Secondary (million L day^{-1})	Advanced-2nd (million L day^{-1})	(million L day^{-1})
Delta Sacramento, Stockton, Vacaville, Tracy, Manteca/Lathrop, West Sacramento, Brentwood, Discovery Bay, Mountain House	692.5	142.4	834.9
Sacramento Basin Roseville- Dry Creek, Chico, Redding-Clear Grove, Davis, Roseville-Pleasant Grove, Yuba City, Oroville, Woodland, Redding-Stillwater, Lincoln, Placerville, Grass Valley, Olivehurst, Placer County, University of California Davis, Red Bluff, Anderson, Linda, Auburn, Willows, Corning	100.2	183.3	283.5
San Joaquin Basin Modesto, Turlock, Merced, Lodi, Deer Creek, Galt El Dorado Irrigation District El Dorado Hills	175.0	40.0	215.0
Total Discharge	967.7	365.6	1, 333.3
Percent of Total	72.6	27.4	100

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1204 **Figures**

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1207 **Fig. captions**

1208 **Fig. 1** Locations within the San Francisco watershed of the legally defined Delta (delineated by
1209 bold black line), municipal wastewater facilities, maximum extent of cyanobacteria (*Microcystis*
1210 *aeruginosa*) blooms, and locations of intermittent, acute toxicity of field-collected surface waters
1211 to larval fish (fathead minnows, *Pimephales promelas*), invertebrates (water fleas, *Ceriodaphnia*
1212 *dubia*), or freshwater algae (*Selenastrum capricornutum*) in laboratory tests, based on standard
1213 U.S. Environmental Protection Agency protocols, from 2008 to 2009

1214 **Fig. 2** Abundance indices of pelagic fishes in the Delta from 1967 to 2009. The indices are based
1215 on numbers of fish captured during midwater trawls from September to December, but do not
1216 represent actual abundances. Note differences in scale of y-axes. (Data source: California
1217 Department of Fish and Game, http://www.delta.dfg.ca.gov/data/mwt/charts.asp)

1218 **Fig. 3** Conceptual model of how phenology and toxic mechanisms potentially affect four pelagic
1219 fishes in the San Francisco Estuary at specific life stages. Turbidity and conductivity increase
1220 from east (Sacramento-San Joaquin Delta) to west (San Francisco Bay). All life stages are
1221 affected by hydrologic changes in turbidity and salinity, impaired water quality from stochastic
1222 rainfall events, and wastewater inputs that impair habitat quality.

1223 **Fig. 4** Deviation from annual mean for a) average maximum and b) average minimum water
1224 temperatures. Daily water temperatures were collected at 15-min intervals at Stockton, Antioch,
1225 and Rio Vista, California between 1984 and 2007. (Data from www.iep.water.ca.gov)

1226 **Fig. 1**

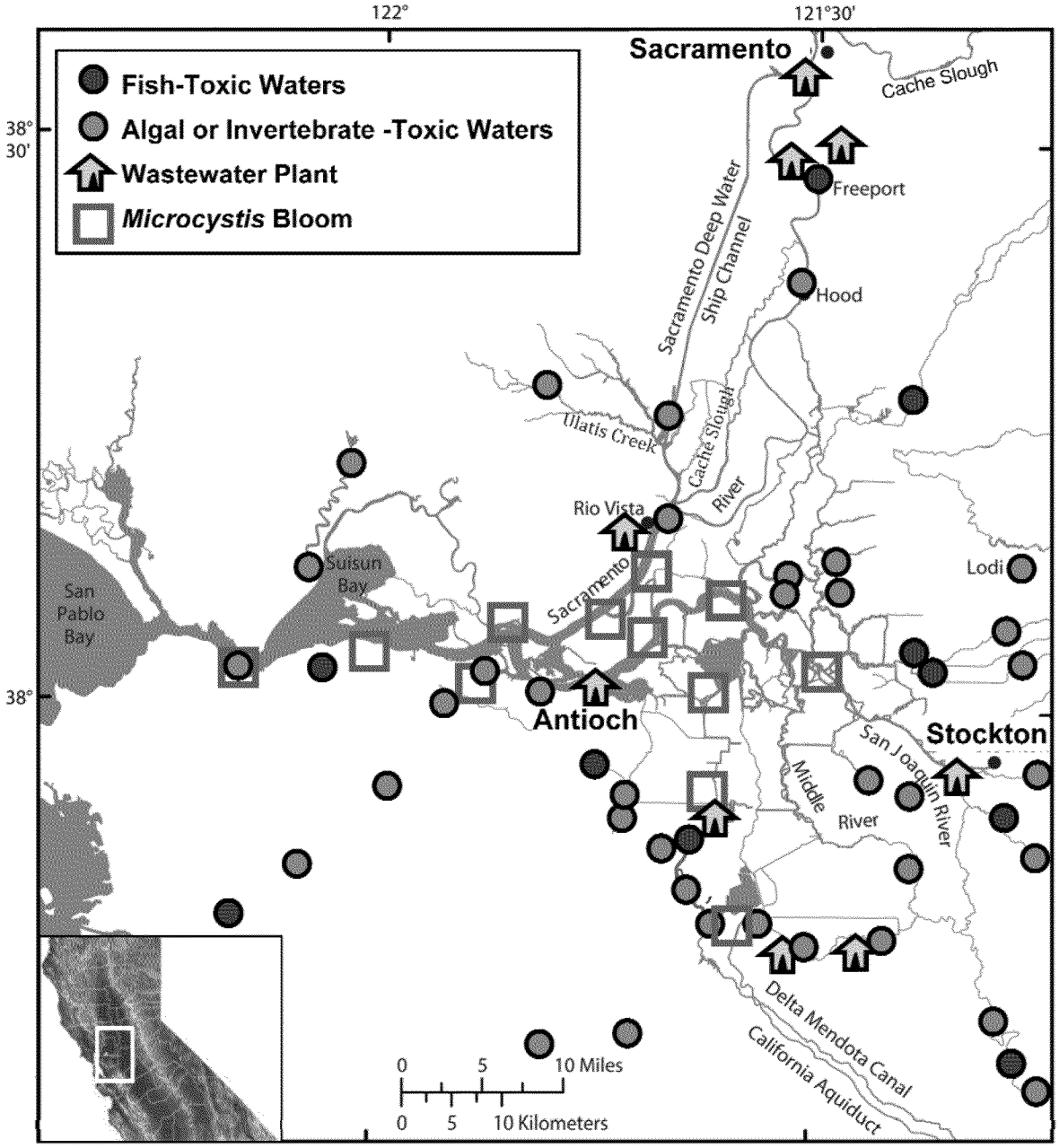

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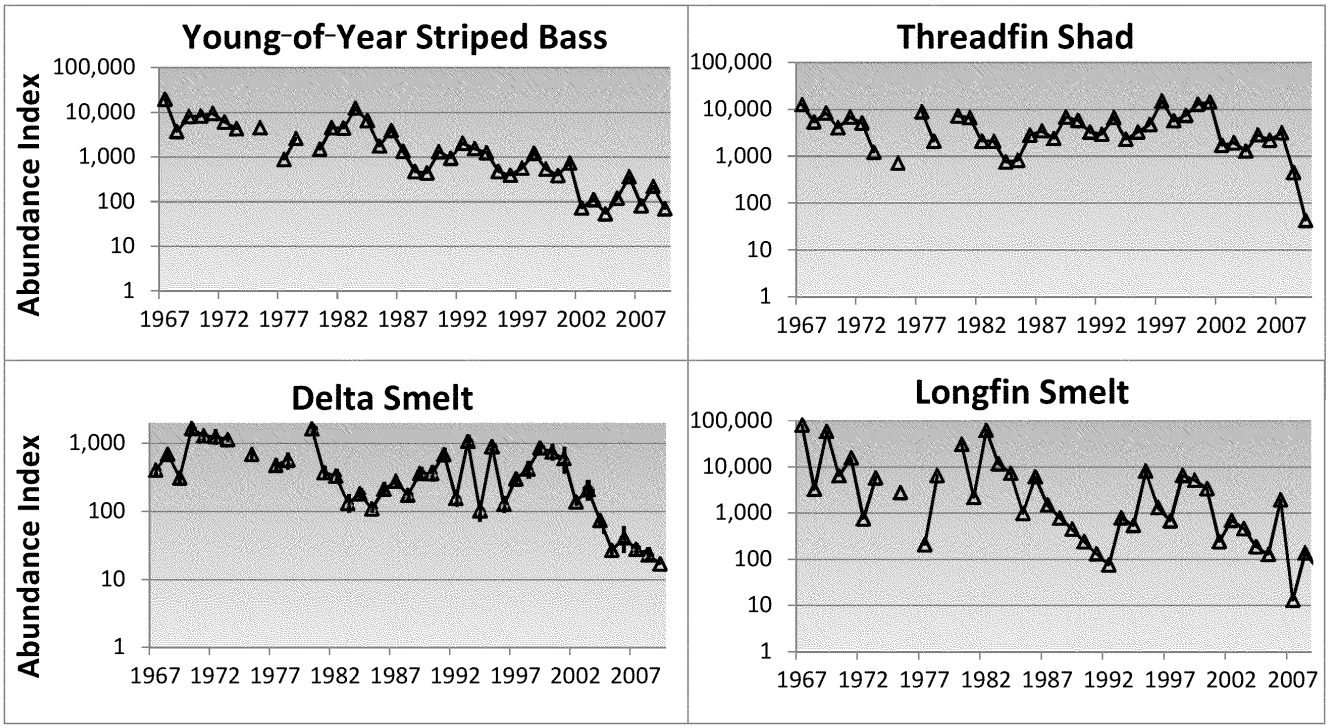
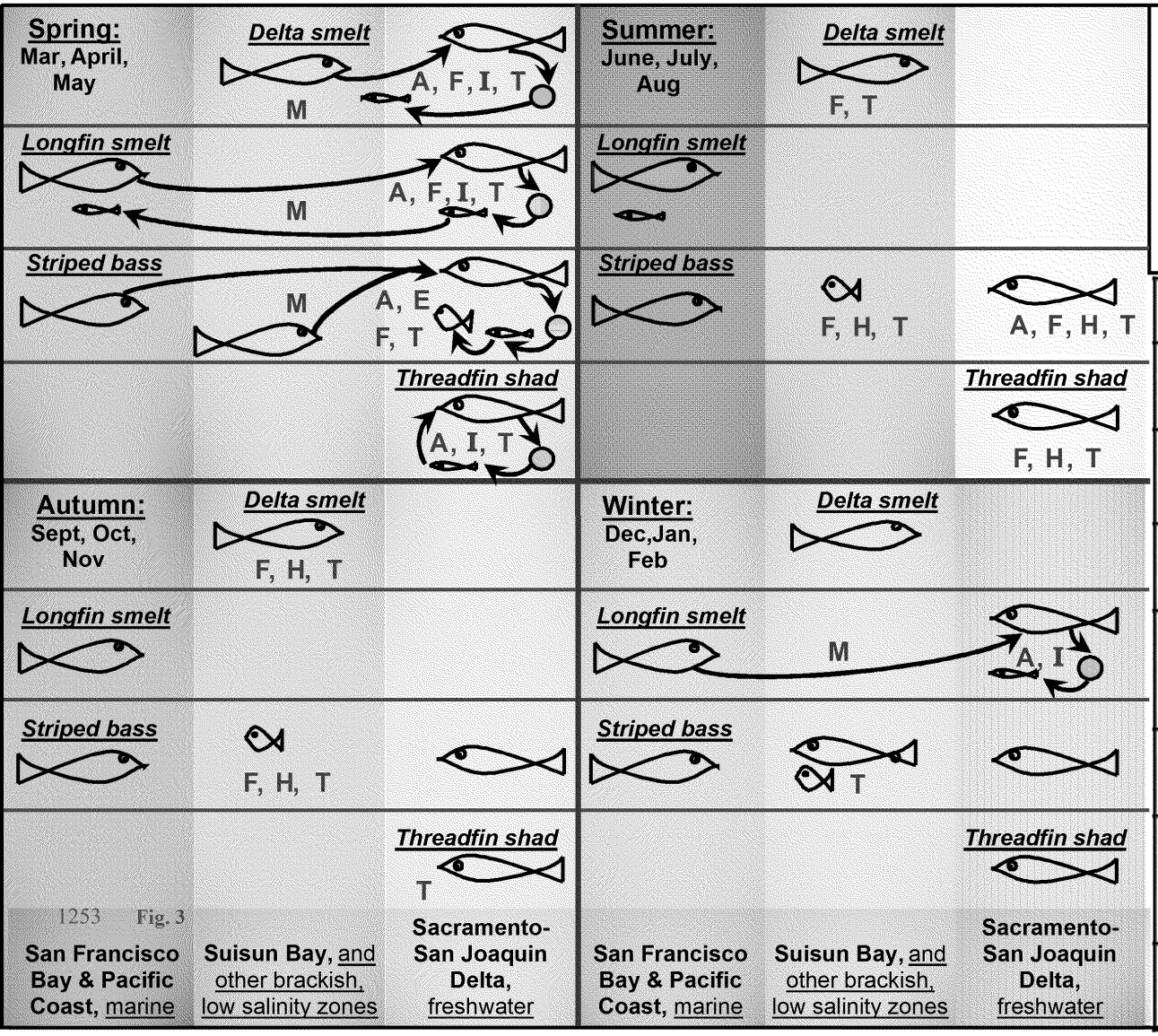
Fig. 2


Young-of-Year Striped Bass
Abundance Index
100,000
10,000
1,000
100
10
1
1967 1972 1977 1982 1987 1992 1997 2002 2007
Threadfin Shad
100,000
10,000
1,000
100
10
1
1967 1972 1977 1982 1987 1992 1997 2002 2007
Delta Smelt
Abundance Index
1,000
100
10
1
1967 1972 1977 1982 1987 1992 1997 2002 2007
Longfin Smelt
100,000
10,000
1,000
100
10
1
1967 1972 1977 1982 1987 1992 1997 2002 2007



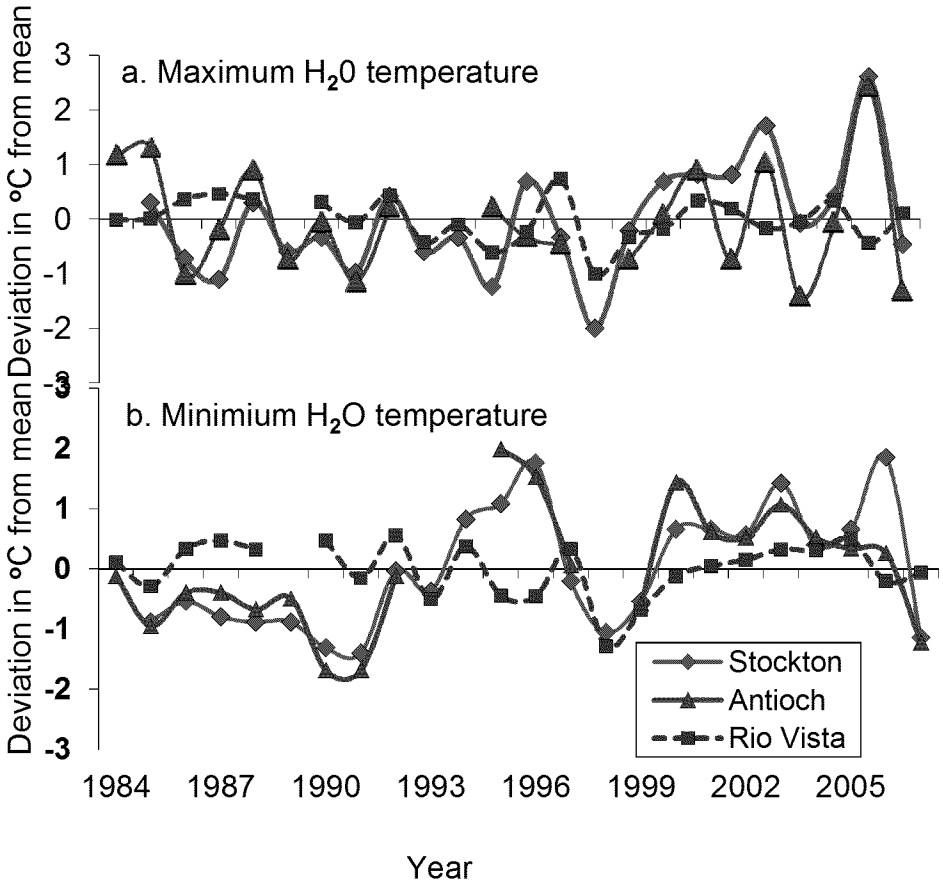
1253 Fig. 3

Life stages

- Adult
- Egg
- Larvae or fry
- 0-age Bass

Toxic Effects	
A	Acute toxicity to larvae and juveniles
F	Direct or indirect food limitation
E	Maternal transfer of toxins to eggs
I	Impaired behavior, disease susceptibility
H	Harmful algal blooms
M	Migratory release of toxins from fat reserves
T	Temperature effects on toxic threshold

Fig. 4


a. Maximum H_20 temperature
b. Minimium H_2O temperature
Deviation in °C from mean
Deviation in °C from mean
3
2
1
0
-1
-2
-3
1984
1987
1990
1993
1996
1999
2002
2005
Year
Stockton
Antioch
Rio Vista